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Review

Biophysical risks to carbon sequestration and storage in Australian drylands

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ABSTRACT

Carbon abatement schemes that reduce land clearing and promote revegetation are now an important component of climate change policy globally. There is considerable potential for these schemes to operate in drylands which are spatially extensive. However, projects in these environments risk failure through unplanned release of stored carbon to the atmosphere. In this review, we identify factors that may adversely affect the success of vegetation-based carbon abatement projects in dryland ecosystems, evaluate their likelihood of occurrence, and estimate the potential consequences for carbon storage and sequestration. We also evaluate management strategies to reduce risks posed to these carbon abatement projects. Identified risks were primarily disturbances, including unplanned fire, drought, and grazing. Revegetation projects also risk recruitment failure, thereby failing to reach projected rates of sequestration. Many of these risks are dependent on rainfall, which is highly variable in drylands and susceptible to further variation under climate change. Resprouting vegetation is likely to be less vulnerable to disturbance and have faster recovery rates upon release from disturbance. We conclude that there is a strong impetus for identifying management strategies and risk reduction mechanisms for carbon abatement projects. Risk mitigation would be enhanced by effective co-ordination of mitigation strategies at scales larger than individual abatement project boundaries, and by implementing risk assessment throughout project planning and implementation stages. Reduction of risk is vital for maximising carbon sequestration of individual projects and for reducing barriers to the establishment of new projects entering the market.

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Contents

1. Introduction 103
2. Unplanned fire 104
 2.1. Likelihood of fire occurrence 104
 2.2. Consequences of unplanned fire for carbon storage 104
  2.2.1. Carbon emissions 104
  2.2.2. Post-fire carbon sequestration 105
3. Drought and heat stress 105
 3.1. Likelihood of drought and heat stress 106
 3.2. Consequences of drought and heat stress for carbon storage and sequestration 106
4. Grazing by domestic and wild herbivores 106
 4.1. Factors influencing grazing pressure 106
 4.2. Consequences of grazing for carbon storage and sequestration 106
5. Recruitment failure 107
 5.1. Presence of a seedbank 107

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5.2. Germination and establishment 107
5.3. Competition 107
6. Climate change 107
6.1. Potential consequences of climate change 108
7. Relative importance of identified biophysical risks to carbon abatement projects 108
8. Management strategies and risk reduction mechanisms 108
8.1. Unplanned fire 108
8.1.1. Fuel loads 108
8.1.2. Fire suppression 108
8.2. Mortality and recovery following disturbance 109
8.3. Grazing by domestic and wild herbivores 109
8.4. Recruitment failure 109
8.5. Climate change 109
9. Conclusions 109
Acknowledgements 110
References 110

1. Introduction

Market-based incentives to protect vegetation and revegetate degraded landscapes are an important component of carbon abatement policies around the world. For example, the UN's REDD+ programme now operates across 64 developing countries (UN, 2016), while Australia's Emissions Reduction Fund has established AUD$2.55 billion for abatement (Australian Government, 2014). In addition to providing a mechanism for offsetting carbon emissions, these projects offer a range of social and environmental co-benefits, including diversified income streams for rural landholders, job creation, improvements to biodiversity, reduced erosion and nutrient runoff, concomitant improvements to water quality in freshwater and marine environments, pollination services, livestock shelter, salinity control and increased amenity values (Cunningham et al., 2015). However, these projects may also have adverse impacts, for example forestry activities that impact on water availability or biodiversity conservation, and limiting land access for agricultural production (Australian Government, 2014).

For vegetation-based carbon abatement projects to succeed in mitigating climate change, they must store carbon over an extended time period, otherwise any benefits accrued may be lost. However, there are a number of biophysical risks to maintaining these long-term carbon stores, including fire, drought and heat stress, grazing by livestock and wild herbivores, recruitment failure after active regeneration, and changes in climate. These risks can result in reduced rates of sequestration and the release of stored carbon back to the atmosphere (Galik and Jackson, 2009). For example, wildfires are estimated to reduce the annual terrestrial carbon uptake by 0.32 Pg C yr^{1}, which accounts for around 20% of the total annual terrestrial carbon sink in a world without fire (Yue et al., 2015). In 2003, drought and heatwaves in Europe were estimated to reduce ecosystem gross primary productivity by 30% and resulted in net carbon emissions of 0.5 Pg C yr^{1} (Ciais et al., 2005). These risks not only affect the environmental and economic value of existing carbon abatement projects, but may also inhibit their uptake by additional landholders, effectively reducing carbon abatement potential. Thus, understanding potential risks to carbon abatement projects is crucial to their success in mitigating climate change.

Here, we undertake a review of risk factors for vegetation-based carbon abatement initiatives, with a focus on Australian drylands. Drylands are characterised by infrequent, highly variable rainfall, and are defined as regions with an aridity index (which is the ratio of mean annual precipitation to potential evapotranspiration) below 0.65 (UNCCD, 2000). We focus on drylands because they offer considerable opportunities for carbon sequestration due to their extensive land area, covering around 30% of Earth's land surface (Lal, 2004) and 70% of Australia (Fig. 1; Eamus et al., 2016). In Australia, carbon projects that involve either avoided clearing of vegetation or regeneration of previously cleared or degraded vegetation are concentrated in drylands, with a large number of projects under the Australian Emissions Reduction Fund located in rural properties in the semi-arid region of western New South Wales (Fig. 1). These initiatives offer opportunities to mitigate climate change, rehabilitate degraded landscapes and drive economic stability (Dean et al., 2012, 2015).

Assessment of the risks to carbon abatement projects requires elucidation of both the likelihood and consequences of identified risk factors for carbon storage and sequestration. In this review we identify and examine each of the following risk factors: (i) unplanned fire; (ii) drought and heat stress; (iii) grazing by livestock and wild herbivores; (iv) factors leading to recruitment failure; and (v) climate change. For each risk factor we evaluate the likelihood of occurrence and the potential consequences for carbon storage and sequestration. Finally, we (vi) identify management strategies and

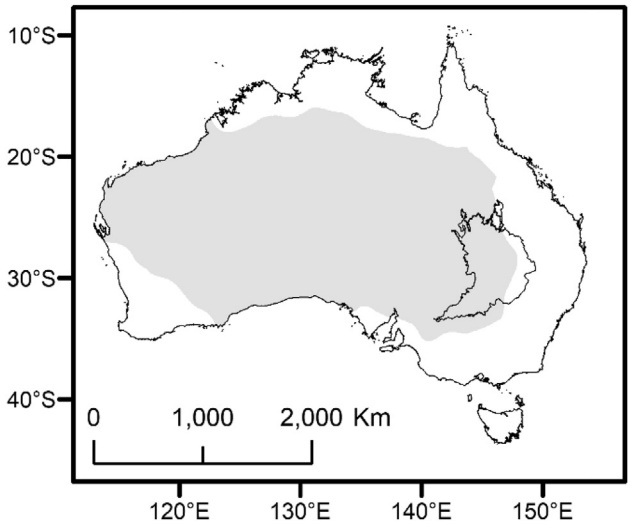

Fig. 1. Location of drylands in Australia, with a majority of carbon farming projects under the Australian Emissions Reduction Fund located within the bioregions bounded by the black polygon.

risk reduction mechanisms that mitigate against these unplanned risk factors.

2. Unplanned fire

2.1. Likelihood of fire occurrence

Fire is a major disturbance across many vegetation types globally. While fire is a natural process over large areas of the Australian continent, it may hamper efforts to enhance sequestration of carbon. In drylands, primary production, and therefore fuel, is limited by rainfall and there is often insufficient biomass or connectivity between fuels for fire propagation (Turner et al., 2011). Fire occurrence generally tracks periods of above-average rainfall when there is an increase in plant growth; a process that drives increased fuel loads and connectivity between fuels (Fig. 2a; Turner et al., 2011). In drylands, rainfall is highly variable from year to year, particularly in Australia (van Etten, 2009), hence the occurrence of fire is also highly variable. For example, in mulga woodlands, which cover 20–25% of the Australian continent and dominate arid and semi-arid landscapes (Eamus et al., 2016), fire return intervals range from 3 to 52 years (Ward et al., 2014).

Where carbon projects involve regeneration of vegetation, fuel loads will increase over time, and this may have implications for the intensity and rate of spread of fire across a landscape (Jenkins et al., 2016). However, this is dependent on the matrix of fuel loads across a landscape (Collins et al., 2015; Jenkins et al., 2016). For example, in a catchment in eastern Australia vegetated with temperate *Eucalyptus* forests and woodland, Collins et al. (2015) found that simulated increases in woody vegetation from environmental plantings increased modelled fire sizes if there were low fuel loads (2 t ha^{-1}) across pastures in the catchment area. However, if fuel loads in pastures were moderate (4.5 t ha^{-1}) or high (7 t ha^{-1}), there was either little change in mean fire size, or else fire sizes declined in response to environmental plantings. Similarly, Jenkins et al. (2016) found that revegetation does not always increase rates of fire spread across a landscape, because modelled flame heights were sometimes higher in pastures than environmental plantings, depending on the weather conditions. Thus, the risk of fire in carbon project areas is highly dependent upon the fuel loads and fuel connectivity in the surrounding landscape, especially the fuel loads of pastures.

If fuel loads are sufficient to propagate fire, there are still three other factors, or "switches", required for a fire to occur: (1) availability of fuel to burn, (2) fire weather, and (3) ignition (Bradstock, 2010). Fuels are available to burn when they are sufficiently low in moisture content for fire ignition to occur. In arid and semi-arid landscapes this pre-condition for fire is frequently met, as is the pre-condition of fire weather, i.e. hot, dry and windy conditions (Turner et al., 2011). Thus, the limiting pre-condition for fire in arid and semi-arid environments is fuel load and fuel connectivity, in contrast to temperate environments where fuel dryness and fire weather are the key limiting factors for fire (Bradstock, 2010). The final factor required for fire is ignition, and this usually occurs from lightning during dry summer storms or anthropogenic sources (Bradstock, 2010).

2.2. Consequences of unplanned fire for carbon storage

Terrestrial carbon stores are affected by fire in three main ways: (1) carbon is emitted to the atmosphere by combustion during a fire; (2) fire-killed plants slowly decompose and release carbon; and (3) carbon accumulates following fire as the vegetation regenerates (Williams et al., 2012). Fire may additionally indirectly affect carbon stores by increasing the risk of erosion, which can transport substantial amounts of soil organic carbon, depending on the topography of the burnt area (Smith et al., 2011). Fire additionally affects soil carbon through effects on soil microbial communities and nutrient cycling, with enhanced respiration often observed due to soil heating (Munoz-Rojas et al., 2016).

2.2.1. Carbon emissions

Carbon emissions from fire, predominantly as carbon dioxide, carbon monoxide and methane, occur both at the time of the fire and over successive years, and vary according to vegetation type (Sommers et al., 2014). Emissions from annual dry season fires in mesic savannas, which are distributed across Australia's northern drylands, have been estimated at 1.5–3 t C ha^{-1} yr^{-1} (Beringer et al., 2007). To put this in context, net ecosystem productivity (NEP) has been estimated as 3.5–5 t C ha^{-1} yr^{-1} (Beringer et al., 2007). In higher productivity temperate eucalypt forests, where fire return intervals are longer (typically in the order of decades), carbon emissions from fire have been estimated at upwards of 11 t C ha^{-1}, while NEP has been estimated at up to 6.7 t C ha^{-1} yr^{-1} under conditions of average rainfall (Williams et al., 2012).

Following fire, carbon emissions continue as fire-killed plants slowly decay. These emissions may be much higher than pyrogenic emissions but occur over years to decades. For example, in western US forests, carbon emissions from fire have been estimated at 4 Tg C yr^{-1} but the amount of biomass transferred from live to dead and decomposing carbon pools by fire was estimated at 10.5 Tg C yr^{-1} (Ghimire et al., 2012). Overall carbon emissions are a function of

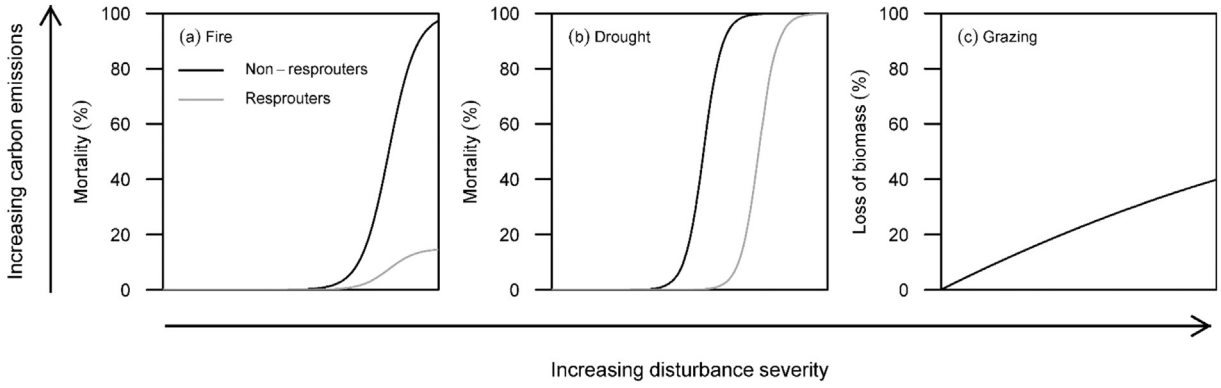

Fig. 2. Conceptual models of the risk to carbon storage due to (a) fire; (b) drought and associated heat stress; and (c) grazing. The conceptual model for fire is modified from logistic regression models of mortality due to fire from Catry et al. (2013); the conceptual model for drought is derived from curves of xylem vulnerability to cavitation (Mitchell et al., 2016; Zeppel et al., 2015); and the conceptual model for grazing is derived from a meta-analysis of plant biomass responses to grazing in Australian rangelands (Eldridge et al., 2016).

mortality rates, which generally increase as fire severity increases (Fig. 2a; Williams et al., 2012). However, some species have low mortality rates through their ability to resprout following disturbance, even after high severity fire resulting in complete defoliation (Fig. 2a). Thus, the capacity of species to resprout following disturbance is important in determining fire-driven mortality rates and subsequent carbon emissions.

2.2.2. Post-fire carbon sequestration

Following fire, productivity declines due to loss of biomass, and vegetation often changes from a net sink to a net source of carbon. Rates of post-fire vegetation recovery, and hence recovery of carbon stores, are dependent on the ecological responses to disturbance of constituent species. Seedling germination may be triggered through heat and chemical cues, opening serotinous seed capsules (Noble and Slatyer, 1980). Resprouting species can recover relatively quickly following fire in comparison to species regenerating through seed (Bell and Pate, 1996). Thus, for resprouters, this strategy may lead to a more rapid accumulation of carbon stores than vegetation regenerating from seed (Fig. 3; Adams et al., 2012; Nolan et al., 2015). Vegetation communities in the study area include both obligate seeding (i.e. fire-killed, regenerate from seed) and resprouting vegetation types. *Callitris* forests and woodlands and mulga (*Acacia*) woodlands are killed by fire and regenerate via seeds during high intensity fire (Cohn et al., 2011; Ward et al., 2014), whereas *Eucalyptus* woodlands resprout following fire (Clarke et al., 2015).

Rates of post-fire accumulation of carbon stores also depend on seedling germination and survival. Successful recruitment of seedlings is important even for resprouting vegetation types, because there is often some post-fire mortality and seedling germination in these species (Noble and Slatyer, 1980). If the interval between fires is too short, some species may not have reached reproductive maturity and will be abent from the seedbank. Conversely, if the interfire interval is too long, this may also affect seed banks through a decline in fecundity with age. For example, for *Callitris verrucosa*, maturation occurs at 10–15 years and fecundity declines from *c*. 80 years (Bradstock and Cohn, 2002). Thus, a fire return interval of less than 20–25 or greater than 80 years may lead to a decline in population numbers of this species. In some species the seed bank can also be stored in soil, however for arid environments high seed predation strongly limits the soil seed bank of woody species (Wright and Clarke, 2009). Optimal fire-return intervals are highly uncertain for many vegetation communities in Australia's drylands (Noble, 1984), this may be due in part to the infrequent, irregular nature of fire in these environments.

3. Drought and heat stress

Drought and heat stress can occur independently and both can affect rates of carbon sequestration. However, these two risk factors

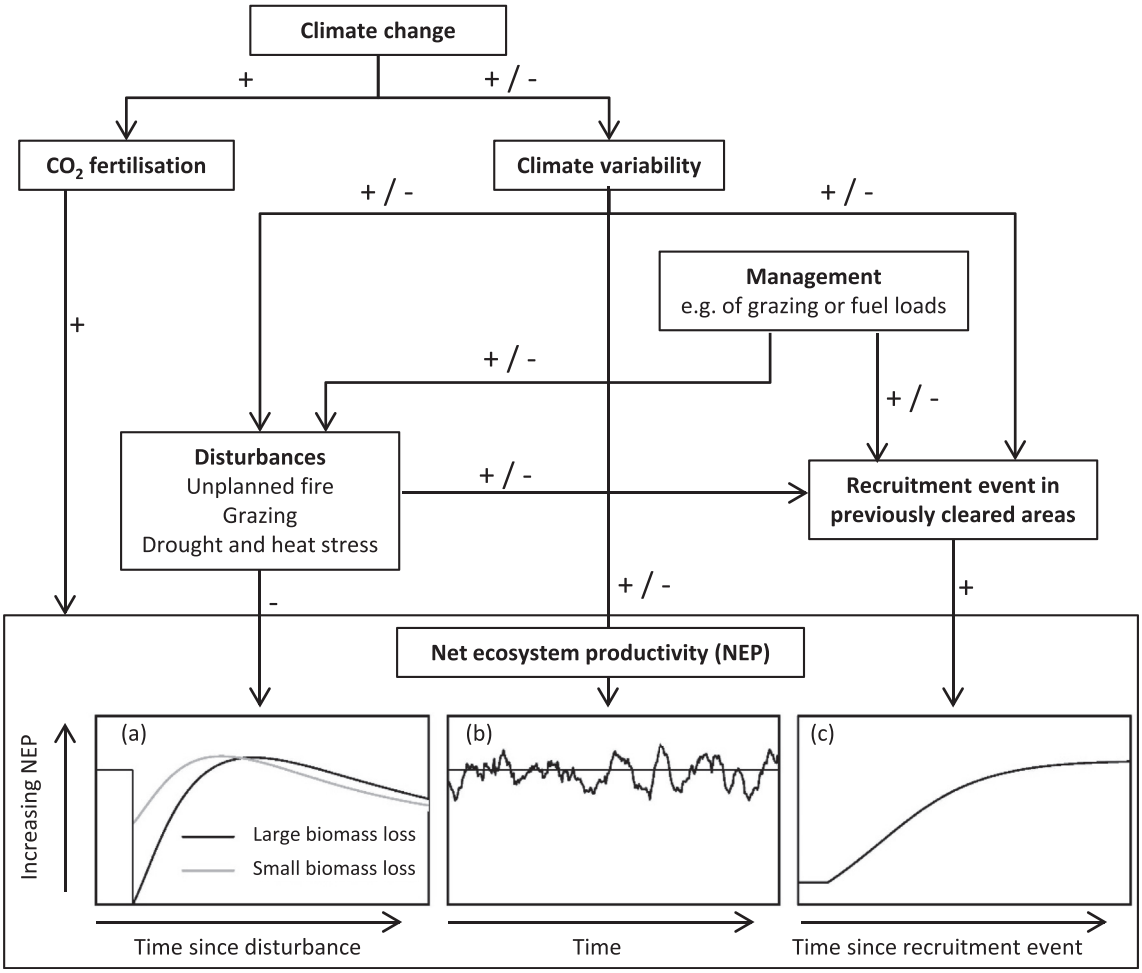

Fig. 3. Conceptual model showing the relationships between biophysical risk factors and their effects on net ecosystem productivity (NEP), including (a) NEP recovery following disturbance; (b) natural variability in NEP due to climate; and (c) increasing NEP following a recruitment event; +/− symbols represent positive or negative effects on NEP.

often co-occur (Kiem et al., 2016). Given the close coupling of these two risk factors, we discuss their effects on carbon storage and sequestration together.

3.1. Likelihood of drought and heat stress

Precipitation in Australia is influenced by four major climate drivers: the El Niño-Southern Oscillation in the Pacific Ocean; Pacific Decadal Oscillation; Indian Ocean dipole; and Southern Annular Mode in the Southern Ocean (Kiem et al., 2016). Individually, each system is associated with cycles of drought or flood, but these systems can also interact with each other. For example, drought in 2009 and an exceptional rainfall year of 2011 have been linked to interactions between these climate systems (Cleverly et al., 2016). Although there is increasing understanding of the mechanisms causing drought in Australia, e.g. changes in sea-surface temperatures and pressure anomalies, there is currently limited ability to forecast when a drought will occur or how long it will last. This is partly due to the complex interactions among climate drivers, and difficulties in defining and monitoring drought due to the variety of temporal and spatial scales over which drought occurs (Kiem et al., 2016). Heatwaves are also difficult to forecast. Higher temperatures are often associated with drought, but the cause of this is not straightforward. Drought increases the likelihood of lower cloud cover and actual evapotranspiration, which lowers evaporative cooling and, as a result, increases plant temperatures. Conversely, higher temperatures may increase the probability of drought by increasing potential evapotranspiration (Kiem et al., 2016).

3.2. Consequences of drought and heat stress for carbon storage and sequestration

Even though arid and semi-arid vegetation is adapted to low rainfall, these communities still respond to cycles of wet and dry. For example, the largest recorded global land carbon sink of 4.1 Pg C yr^{-1} occurred during the wet year of 2011. In comparison, the average land carbon sink is 2.6 Pg C yr^{-1}, with over half of this increase attributed to exceptional precipitation, and subsequent productivity, across Australia's semi-arid regions (Poulter et al., 2014). However, this increase in the carbon sink across semi-arid regions of Australia was transient, with productivity rapidly diminishing in the following years (Ma et al., 2016).

Dryland plants typically exhibit a range of strategies for surviving under low rainfall. Some plants avoid aridity by accessing groundwater, usually through deep roots (Nolan et al., 2017a), while others may have numerous morphological and physiological adaptations to survive under conditions of low soil water availability (McDowell et al., 2008). Plants can respond to drought via a number of morphological and physiological mechanisms. This includes shedding leaves and closing stomata, resulting in reduced rates of transpiration, and hence photosynthesis (Tardieu and Simonneau, 1998; Nolan et al., 2017b). Prolonged drought can lead to enhanced rates of mortality and subsequent carbon emissions (Fig. 2b). While mortality is correlated with drought severity, mortality rates can be affected by interacting factors, such as insect infestation, which often co-incide with drought (Anderegg et al., 2015).

The effects of drought on carbon storage and sequestration depends on how different plants respond to disturbance. For resprouting species, mortality may be delayed compared to non-resprouting species (Fig. 2b; Zeppel et al., 2015). Further, ecosystem recovery is likely to be more rapid following release from drought, due to the rapid recovery of foliage in resprouting plants (Fig. 3; Zeppel et al., 2015). Thus, the impacts of drought on carbon stores are likely to be lower in vegetation dominated by resprouting species.

4. Grazing by domestic and wild herbivores

Domestic livestock in Australian drylands include sheep (*Ovies aries*), cattle (*Bos taurus*, *Bos indicus*), and goats (*Capra hircus*) (Dean et al., 2015), with goats and cattle also occurring as wild herbivores. Other common wild herbivores include kangaroos (*Macropus* spp.), rabbits (*Oryctolagus cuniculus*), feral horses (*Equus caballus*), feral donkeys (*Equus asinus*), camels (*Camelus dromedaries*), and deer (*Cervus* spp., *Axis* spp.) (Eldridge et al., 2016). Invertebrates can also contribute to grazing (Chapman et al., 2003), but are not explicitly considered here.

4.1. Factors influencing grazing pressure

Herbivore density is limited by available resources, particularly rainfall, and therefore soil moisture, which drives primary productivity. For example, annual wool production (measured as kg/head/year) is correlated with mean annual rainfall which affects the quality and availability of forage (Freudenberger et al., 1999). Similarly, kangaroo densities are known to fluctuate widely in relation to mean annual rainfall (Letnic and Crowther, 2013). The distribution of domestic and wild herbivores is often driven by the availability of watering points (Fensham and Fairfax, 2008), though kangaroos are not strongly water-focused (Montague-Drake and Croft, 2004). The density of large herbivores may also be moderated by the presence of predators such as dingoes (*Canis dingo*) (Johnson and Wallach, 2016; Letnic and Crowther, 2013), although most livestock susceptible to predation by dingoes are protected by dingo-proof fencing.

4.2. Consequences of grazing for carbon storage and sequestration

The effects of grazing on carbon storage and sequestration vary with species, grazing intensity, and climate (Conant et al., 2017; Eldridge et al., 2016). In a review of 217 studies on the effects of grazing in Australian drylands, Eldridge et al. (2016) found that livestock grazing generally reduced above-ground biomass, by 40% on average; with larger effects in drier environments. Eldridge et al. (2016) also found that increasing livestock density resulted in larger reductions in biomass (Fig. 2c); with cattle having greater effects on biomass than sheep, but synergistic effects when both grazed together. Goats also have larger effects on biomass than sheep, and at high stocking rates goats can cause widespread shrub mortality (Harrington, 1979). In contrast to above-ground biomass, the effect of livestock grazing on below-ground biomass is variable. In a global review of grazing effects across 276 sites across a range of climate zones, Milchunas and Lauenroth (1993) found that although livestock grazing generally reduced above-ground biomass, both increases and reductions in root biomass and soil organic carbon were observed with grazing. Increased allocation to root biomass allows for faster recovery of photosynthetic material lost to grazing, but is likely constrained by nutrient and water availability (Milchunas and Lauenroth, 1993). The relationship between increasing grazing intensity and soil carbon is, however, complex, with declines in carbon at low and high grazing contrasts, but neutral effects under moderate grazing (Allen et al., 2013; Eldridge et al., 2017; Maestre et al., 2016). Grazing can also affect soil organic carbon storage by increasing the risk of erosion (Eldridge et al., 2016, 2017); through additions of animal excreta, which may enrich soil organic carbon (Hunt et al., 2016); or through livestock carbon emissions (Dean et al., 2012).

There are some exceptions to the trend of declining above-

ground biomass with increasing livestock grazing. Notably, shrub encroachment, or "woody thickening", has been observed across drylands globally and is due to a combination of factors, including grazing (van Auken, 2009). Livestock grazing can reduce grassy and herbaceous biomass, thereby reducing competition and fire frequency, which favours woody species (van Auken, 2009). Shrub encroachment may also be due to increases in CO_2 and N deposition, although these are likely to be less important than grazing (van Auken, 2009). Shrub encroachment is likely to increase rates of carbon sequestration across drylands (Daryanto et al., 2013; Eldridge and Soliveres, 2014). Effects of grazing on woody encroachment are likely to be the legacy effects of historic overgrazing up to a century ago.

There has been less research on the effects of grazing by wild herbivores on carbon sequestration, than effects by domestic herbivores. Many studies on the effects of livestock grazing on biomass also include wild grazers, thus potentially confounding any effects of wild grazers from those of livestock (Eldridge et al., 2016). However, domestic livestock and wild grazers have differing dietary preferences (Dawson and Ellis, 1994, 1996) and may therefore differ in their effects on carbon sequestration. In general, macropods (e.g. kangaroos and wallabies) eat more grasses and less woody plants than sheep (Dawson and Ellis, 1994), and so are unlikely to have similar effects on above-ground biomass. Similarly, goats eat more woody biomass than macropods (Dawson and Ellis, 1996). Compared to domestic livestock, sites grazed by macropods and rabbits tend to have a greater vegetation cover and abundance of perennial vegetation (Fensham and Skull, 1999). Indeed, recent research on grazing effects by different herbivores in the semi-arid woodlands of eastern Australia has shown that kangaroos had no effects on plant community composition (Travers et al., 2017). The effects of wild herbivores on carbon sequestration will therefore likely differ from those of European livestock. The effects will vary depending on the species of wild herbivore. Grazing can also affect other aspects of ecosystem functioning which then influences rates of carbon sequestration. For example, kangaroos have been shown to be important local recyclers of energy (Iles et al., 2010), a process that may influence sequestration rates. Current knowledge of the importance of these types of functions is poor.

5. Recruitment failure

For carbon abatement projects that involve revegetation or regeneration, projects may fail to realise maximum potential rates of carbon sequestration if plants fail to germinate or establish. An exception to this is for projects that involve direct planting of seedlings, though these projects are still vulnerable to failure in seedling establishment. Here we discuss factors that may limit or prevent successful revegetation or regeneration.

5.1. Presence of a seedbank

Regeneration and natural revegetation projects require the presence of a viable seed bank. Seed production in drylands typically occurs in pulses in response to above-average rainfall. For example, seed masting, the production of large seed crops, occurs in mulga, a group of 12 closely related *Acacia* species which cover 20% of continental Australia (Preece, 1971). Masting in mulga usually only occurs following above-average rainfall both in summer and the following winter (Preece, 1971). Irrespective of seed production strategy, seed storage in the soils of Australian drylands is limited due to high rates of predation, often by ants (Wright and Clarke, 2009). Some species, such as *Eucalyptus* spp. can avoid predation by storing seeds in canopy (serotinous) seedbanks (Wellington and Noble, 1985). Given the short residence time for seeds in soils, there is unlikely to be a high diversity of seeds stored in soils that have been managed for agricultural production, and emerging seedlings may face substantial competition from herbaceous species (Semple and Koen, 2006). The exception to this are short-lived herbaceous flora, which can dominate soil seed banks over woody species in arid areas (Wright and Clarke, 2009). Thus, successful natural regeneration of agricultural land, particularly of woody species, will depend in large part on proximity to a seed source, such as remnant vegetation or scattered trees.

5.2. Germination and establishment

Recruitment depends on the presence of seeds and successful germination. In many species germination may be delayed until the occurrence of an abiotic trigger, such as above-average rainfall or fire. For example, in serotinous species such as mallee and *Callitris*, seed release is triggered by fire (Bradstock and Cohn, 2002; Wellington and Noble, 1985); while mulga requires high severity fire, or temperatures exceeding 80 °C, for substantial rates of germination to occur (Wright et al., 2016). In drylands, the timing and amount of rainfall is also important for seed germination, emergence and seedling survival (Cohn and Bradstock, 2000; Fehmi et al., 2014). For species with fire-triggered germination, soil moisture conditions are often ideal for germination because fire generally follows above-average rainfall. Given that rainfall in drylands is highly variable (van Etten, 2009), germination rates and seedling mortality are likely to vary substantially across years.

5.3. Competition

Competition also has important implications for revegetation success and carbon sequestration. In younger revegetation projects, competition can increase mortality rates of some species (Rinella et al., 2015). Woody species may be particularly vulnerable to this competition because they are under-represented in soil seed banks compared with herbaceous or grassy species (Rinella et al., 2015), and the survival rate of woody species often declines with increasing grassy biomass (Hild et al., 2006). However, this has not been tested in Australian semi-arid landscapes. Competition could thus have important consequences for long-term rates of carbon sequestration if trees and shrubs are out-competed by grassy or herbaceous species during the initial stages of revegetation.

If woody species dominate in natural regeneration projects, they may form dense stands of regrowth. This can be common where revegetation occurs on former agricultural soil located close to remnant woodlands or forests. For example, in western New South Wales (Fig. 1), *Callitris glaucophylla* often regenerates on previously cleared land, forming mono-specific stands with stem densities exceeding 1500 stems ha^{-1} (McHenry et al., 2006). Increased densities of woody stems could enhance total carbon storage in the overstorey, but above-ground carbon is greater in systems with a few large trees than those with many small-stemmed trees of low basal area (Eldridge and Wilson, 2003). High density patches may be associated with reduced carbon sequestration over long time scales (Dwyer et al., 2010). This is because high density stands typically have slower rates of overstorey biomass accumulation and reduced understorey biomass due to competition for resources (Dwyer et al., 2010; McHenry et al., 2006).

6. Climate change

Climate change projections for Australia are for higher temperatures, altered rainfall patterns, and increases in extreme events such as heatwaves (IPCC, 2014). Over the next 50 years, western New South Wales (Fig. 1) is likely to see temperature increases of

2–3 °C and higher mean annual rainfall, particularly in the wetter, summer months, with lower rainfall possible in the drier, spring months (Olson et al., 2016). Rainfall intensity is also likely to increase (Westra et al., 2014). Here, we discuss how these climatic changes may affect the identified risk factors for carbon projects.

6.1. Potential consequences of climate change

The consequences of climate change for carbon sequestration in drylands is highly uncertain, owing to the complex interactions among rainfall, temperature, fire, drought, heat stress, and grazing in these environments (Maestre et al., 2016). Given that productivity in drylands is limited by water availability, increases in mean annual rainfall may increase productivity. Additionally, increased atmospheric CO_2 may lead to a "fertilisation effect" that also increases productivity (Fig. 3; Donohue et al., 2013). However, co-occurring increases in temperature are likely to moderate or reverse any gains in productivity due to higher rates of potential evapotranspiration, and therefore lower soil moisture (Bates et al., 2008). Additionally, productivity declines during heatwaves (van Gorsel et al., 2016). Further, recent research indicates that the CO_2 fertilisation effect may not be realised in phosphorus-limited vegetation, which is common across Australia (Ellsworth et al., 2017).

Changing climate will also indirectly affect carbon sequestration through disturbance and increased probability of wind and water erosion. Increased summer rainfall, if it is sufficient to promote an increase in the growth of ephemeral plants, will increase the risk of unplanned fire. However, there is large uncertainty around future projections of fuel loads in semi-arid regions (Clarke et al., 2016). Given that climate change is projected to affect the seasonality of rainfall, there may be an increase in the frequency of drought events. Apart from the direct effects of drought and heat stress on productivity (Section 3), drought can also indirectly affect carbon sequestration by increasing the vulnerability to disease and insect infestation (Carnicer et al., 2011); and may reduce resistance to erosion (Dean et al., 2012).

7. Relative importance of identified biophysical risks to carbon abatement projects

The risk factors discussed here can largely be classified as disturbances, i.e. fire, drought and heat stress, and grazing. These disturbances affect carbon abatement projects through emissions of stored carbon (Fig. 2) and altered rates of net ecosystem productivity as vegetation recovers following disturbance (Fig. 3). Rates of carbon emissions increase with mortality rates and biomass losses, which in turn are a function of disturbance severity (which incorporates both disturbance intensity and duration; Fig. 2). Mortality rates or biomass loss does not respond linearly to increased disturbance severity, and differs among disturbance types and between resprouting or non-resprouting vegetation types. Specifically, potential carbon emissions are much higher following fire and drought than following grazing, with lower mortality rates, and hence lower carbon emissions, in resprouting compared to non-resprouting vegetation types. It is important to note that disturbances may not always result in reduced sequestration over longer time-frames. For example, fire is necessary for the germination, and hence local persistence, of many plant species. Further, the effects of wild herbivore grazing on carbon sequestration are uncertain, and will likely depend largely on interactions with other disturbances.

Rates of net ecosystem productivity are affected by disturbances, in addition to natural climate variability, the occurrence of recruitment events, particularly for revegetation or regeneration projects, and climate change (Fig. 3). Following disturbance, rates of net ecosystem productivity recovery towards pre-disturbance values are faster if overall biomass losses were smaller, i.e. for a lower severity disturbance or for resprouting vegetation types. The effects of climate change on net ecosystem productivity in Australia's drylands is highly uncertain, but is likely to affect the occurrence and severity of disturbance events, and may increase overall rates of NEP through CO_2 fertilisation, as discussed in section 6.1.

8. Management strategies and risk reduction mechanisms

Opportunities to reduce risks to carbon storage and sequestration can involve management strategies before, during, and after event-based risks such as fire, drought and heat stress. Here, we identify management strategies and risk reduction mechanisms for carbon abatement projects.

8.1. Unplanned fire

8.1.1. Fuel loads

Given that fuel loads are the key pre-condition for fire occurrence in drylands, any actions to reduce the risk of fire to carbon stores should focus on fuel load reduction. Management of fuel loads will be most effective following above-average rainfall when fuel loads are high. However, the efficacy of fuel reduction reduces over time. For example, in temperate and Mediterranean forested landscapes, prescribed burning has been shown to be most effective in reducing the fire hazard during the first 2–5 years after treatment (Holsinger et al., 2016). Given the narrow window where prescribed burning can reduce fire spread, it is likely that most fuels would have recovered prior to any subsequent wildfire in drylands, which can have long fire-return intervals, unless prescribed burning occurred regularly. Indeed, Price et al. (2015) found that the effect of prescribed burning on fire activity across bioregions in south-eastern Australia declined with increasing aridity and fire-return intervals. Fuel reduction within pastures may be highly effective at reducing fire risk across a landscape. The rate of spread of fire is typically much faster in grasslands than in forests (Collins et al., 2015). Pasture fuel loads can be reduced through management actions such as mowing or livestock grazing (Davies et al., 2016).

Activities to reduce fuel loads will necessarily reduce carbon storage. This reduction may be offset by reductions in carbon emissions from wildfires, in which case there is an argument that carbon projects should not be penalised for reducing fuel loads. However, research to date suggests that carbon lost from repeated prescribed burning is not necessarily offset by reduced carbon emissions from subsequent wildfires. For example, in dry forests of the Western U.S., prescribed fire or thinning did not affect subsequent wildfire severity, although combining both treatments did (Kalies and Kent, 2016). This suggests that these cumulative carbon losses from prescribed fire are unlikely to be offset by reduced carbon losses from wildfire.

8.1.2. Fire suppression

Fire suppression activities are most likely to be more effective in containing small, low intensity fires where land managers need fewer resources and can act more quickly. Suppression resources include personal protective equipment, fire-fighting equipment, and access to water, which require adequate preparation prior to wildfire occurrence. Preparation also includes, but is not limited to, well-maintained fire breaks, preparation of a written fire plan, consideration of the capacity to defend assets from fire (which includes personal capacity as well as equipment availability and

condition of the grounds), and the availability of a safe place to seek shelter (Penman et al., 2013).

8.2. Mortality and recovery following disturbance

While it may not be possible to prevent high intensity fires, and it is not possible to prevent drought and associated heat stress events, there are management strategies that may potentially mitigate the effects of these disturbances on carbon stores. Following disturbance, grazing may cause mortality of regenerating shrubs or seedlings, thereby affecting the recovery of net primary productivity, but the evidence for this is mixed. For example, grazing exclusion can enhance post-drought recovery of semi-arid shrubs (DeMalach et al., 2014) while low intensity grazing can enhance the growth rate of woody seedlings by reducing competition from co-occurring grassy species (de Villalobos and Pelaez, 2015). However, grazing did not alter the capacity of the arid zone chenopod shrub *Atriplex vesicaria* to recover from drought in arid Australia, compared with ungrazed shrubs (Eldridge et al., 1990). Importantly, there is currently no evidence that grazing exclusion reduces drought-induced mortality of trees in Eucalypt woodlands (Fensham, 1998). Given the mixed evidence of the effect of grazing exclusion on mortality and seedling recruitment, it is currently uncertain whether management strategies related to livestock and wild herbivore densities would prevent carbon losses due to mortality and enhance recruitment following drought.

Thinning may be another useful mitigation action to reduce the risk of drought-induced tree mortality, because thinning typically reduces stand-level transpiration and can therefore enhance soil moisture content (Whitehead et al., 1984). Given this, thinning may potentially mitigate drought stress, and thus enhance productivity in surviving trees. Indeed, this has been demonstrated across a range of forests and woodlands (Sohn et al., 2016). However, to date there has been no research on whether the carbon lost from thinning is offset by enhanced productivity and reduced mortality in surviving trees.

8.3. Grazing by domestic and wild herbivores

Approaches to managing domestic herbivores can broadly be characterised as either 1) low intensity grazing, which aims to minimise impacts on preferred plant species; or 2) short-term, high intensity grazing, which forces livestock to use the entire paddock (Fynn et al., 2017). Both approaches are typically based on some form of livestock rotation among paddocks (Hawkins et al., 2017). Wild herbivores also contribute to total grazing pressure, and can be managed by fencing, culling or preventing access to water. Removal of watering points may reduce densities of wild herbivores, such as goats, but this may not be feasible in a commercial grazing operation. Other alternative strategies, such as predator-friendly farming involving the protection of apex predators such as dingoes, may also reduce the density of wild herbivores, though this has not been widely tested in Australia (Johnson and Wallach, 2016). Overall, the effects of differing management strategies on plant productivity and therefore soil organic carbon, will likely differ across climate gradients, and soil and vegetation types (Fynn et al., 2017; Hawkins et al., 2017). Insufficient grazing may result in increased herbaceous biomass (Dean et al., 2012), increasing the risk of unplanned fire and preventing the recruitment of woody species through competitive interactions.

8.4. Recruitment failure

Mitigation of risks specific to regeneration or revegetation projects can be undertaken both during the planning and at later stages of a project. At the planning stage, selection of sites that are close to existing vegetation or scattered trees, will increase opportunities for recruitment of woody species. Other aspects of project design, such as size and productivity of the site, are also important for determining rates of carbon sequestration (Cunningham et al., 2015). The germination and establishment phases of natural regeneration projects are perhaps associated with the highest risk, because these processes are dependent on rainfall, which is unpredictable across drylands (Fehmi et al., 2014). Adding mulch to soil may mitigate the effects of irregular rainfall on germination and establishment to some extent (Beggy and Fehmi, 2016). However, irrigation is unlikely to achieve desired outcomes because of the long duration that is often required and the risk of high mortality when irrigation is removed (Fehmi et al., 2014). At the germination stage, an additional risk factor is the absence of fire, which may be required to break seed dormancy for some species (Wright et al., 2016). This may be overcome by pre-treating seeds in direct seeding projects, or planting seedlings. Thinning may be beneficial at later growth stages if there is a high density of woody regrowth, though thinning does not always result in greater rates of carbon sequestration (Dwyer et al., 2010; McHenry et al., 2006).

8.5. Climate change

At the paddock scale, species composition is likely to be important in determining responses of carbon storage and accumulation to climate change. In particular, greater plant species diversity and an herbaceous layer dominated by perennial species, rather than annuals, can enhance resistance and resilience to disturbance (McCann, 2000). Diversity in plant functional traits and genetic diversity within species is also likely to be important for ecosystem resilience to climate change (Millar et al., 2007). Thus, a key mitigation action for revegetation type carbon projects is to design projects with high diversity in terms of species composition, functional traits, and within-species genetics. At landscape and regional scales, dryland resilience to climate change may be enhanced by improving connectivity between vegetation patches to enable species movement and gene flow (Millar et al., 2007). Carbon projects offer an important mechanism to facilitate connectivity across large spatial scales, and thus may be critically important in enhancing ecosystem resilience to climate change in highly fragmented landscapes.

9. Conclusions

We have identified a number of biophysical risks to carbon storage in above-ground biomass. At short time scales, disturbances can cause net carbon emissions, but this carbon can potentially be re-sequestered as biomass recovers following disturbance. This highlights the importance of considering carbon abatement potential of vegetation over multi-decadal time-frames. However, there is still considerable uncertainty around the time taken for carbon emitted from disturbances to be re-sequestered in regenerating vegetation for a large number of dryland vegetation types globally. Further, disturbances such as drought and fire are projected to increase under climate change, and the effects of increasing disturbance frequency on NEP trajectories are also uncertain.

We have identified a number of potential management strategies that may reduce the effects of biophysical risks to vegetation in carbon abatement projects. The risks discussed here will almost always operate at spatial scales larger than that defined by an individual project, which is often a paddock. Thus, effective risk management will be aided by co-ordination of management at

landscape and regional scales. Given that these carbon abatement projects are a relatively new enterprise, there is an opportunity to elucidate and account for risk in the design of existing and future carbon projects. Specifically, a risk assessment framework that can be applied to individual projects would aid in optimising carbon sequestration potential, although there are a number of risks associated with knowledge uptake (Prober et al., 2017). Additionally, an assessment of risks could underpin market mechanisms to mitigate risk, for example through insurance products. This would in turn reduce barriers to landholders entering the market.

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